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**Ambient Water Quality
Criteria: Dichloroethylenes**

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DICHLOROETHYLENES

Ambient Water Quality Criteria

Criteria and Standards Division
Office of Water Planning and Standards
U.S. Environmental Protection Agency
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CRITERION DOCUMENT
DICHLOROETHYLENES

CRITERIA

Aquatic Life

1,1-dichloroethylene

For 1,1-dichloroethylene, the criterion to protect freshwater aquatic life, (as derived using procedures other than the Guidelines), is 530 $\mu\text{g/l}$ as a 24-hour average and the concentration should not exceed 1,200 $\mu\text{g/l}$ at any time.

For 1,1-dichloroethylene, the criterion to protect salt-water aquatic life, (as derived using procedures other than the Guidelines), 1,700 $\mu\text{g/l}$ as a 24-hour average and the concentration should not exceed 3,900 $\mu\text{g/l}$ at any time.

1,2-dichloroethylene

For 1,2-dichloroethylene, the criterion to protect freshwater aquatic life, (as derived using procedures other than the Guidelines), is 620 $\mu\text{g/l}$ as a 24-hour average and the concentration should not exceed 1,400 $\mu\text{g/l}$ at any time.

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1978

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Criteria and Standards Div.

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Section 304(a) of the Clean Water Act (33 U.S.C. 1314(a)), requires EPA to publish and periodically update water quality criteria. These criteria are to reflect the latest scientific knowledge on the identifiable effects of pollutants on public health and welfare, aquatic life, and recreation. This report presents water quality criteria for 1,1-dichloro-, cis 1,2-dichloro-, and trans 1,2-dichloroethylene. It presents concentration criteria for the protection of fresh water and saltwater aquatic life. It presents 'safe' concentrations for humans, and in the case of suspect or proven carcinogens, gives various levels of incremental cancer risk. A section 304(a) water quality criterion is a qualitative or quantitative estimate of the concentration of a water constituent or pollutant in ambient waters which, when not exceeded, will ensure a water quality sufficient to protect a specified water use. Under the Act a criterion is a scientific entity, based solely on data and scientific judgment. It does not reflect considerations of economic or technological feasibility, ...

KEYWORDS: *Water quality, *Toxicology, *Water pollution effects(Humans), *Water pollution effects(Animals), *Toxic substances, *Ethylene/dichloro.

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For saltwater aquatic life, no criterion for 1,2-dichloroethylene can be derived using the guidelines, and there are insufficient data to estimate a criterion using other procedures.

Human Health

1,1-dichloroethylene

For the maximum protection of human health from the potential carcinogenic effects of exposure to 1,1-dichloroethylene through ingestion of water and contaminated aquatic organisms, the ambient water concentration is zero. Concentrations of 1,1-dichloroethylene estimated to result in additional lifetime cancer risks ranging from no additional risk to 1 in 100,000 are presented in the Criterion Formulation section of this document. The Agency is considering setting criteria at an interim target risk level in the range of 10^{-5} , 10^{-6} , or 10^{-7} with corresponding criteria of 1.3 $\mu\text{g}/\text{l}$, 0.13 $\mu\text{g}/\text{l}$, and 0.013 $\mu\text{g}/\text{l}$, respectively.

1,2-dichloroethylene

No human health criterion has been developed.

Introduction

The dichloroethylenes consist of the three isomers: 1,1-dichloroethylene (1,1-DCE), cis 1,2-dichloroethylene (cis 1,2-DCE) and trans 1,2-dichloroethylene (trans 1,2-DCE). As of the early 1960's, neither of the 1,2-dichloroethylenes (1,2-DCEs) had wide industrial usage (Patty, 1963). However, annual production of 1,1-DCE now approximates 120,000 metric tons (Arthur D. Little, Inc., 1976). It is used as a chemical intermediate in the synthesis of methylchloroform and in the production of polyvinylidene chloride copolymers (PVDCs). Among the monomers used with 1,1-DCE in copolymer production are vinyl chloride, acrylonitrile, and alkyl acrylates. The impermeability of PVDCs make them useful primarily as barrier coatings in the packaging industry. Polymers with high vinylidene chloride (1,1-DCE) content such as Saran are widely used in the food packaging industry. The heat-seal characteristics of saran coatings make them useful in the manufacture of non-flammable synthetic fiber. 1,1-DCE polymers have also been used extensively as interior coatings for ship tanks, railroad cars, and fuel storage tanks, and for coating of steel pipes and structures (Wessling and Edwards, 1970).

Dichloroethylenes are clear colorless liquids with the molecular formula $C_2H_2Cl_2$ and a molecular weight of 96.95. 1,1-DCE has a water solubility of 2,500 ug/ml, a vapor pressure of 591 mm Hg, and a melting point of -122.1 degrees C. The cis isomer of 1,2-dichloroethylene has a water solubility of 3,500 ug/ml, a vapor pressure of 208 mm Hg, and a melting point of -80.5 degrees C; trans 1,2-dichloroethylene has a

water solubility of 6,300 µg/ml, a vapor pressure of 324 mm Hg and a melting point of -50 degrees C (Patty, 1963; Wessling and Edwards, 1970). The octanol/water partition coefficient for 1,1-DCE was reported as 5.37 (log P = 0.73) by Radding, et al. (1977), indicating it should not accumulate significantly in animals.

1,1-DCE is not known to occur in nature and ambient levels have not been determined. This chemical is expected to be short-lived in water because of its volatilization to the atmosphere (Hushon and Kornreich, 1976). The 1,2-DCEs have been measured in a limited number of U.S. drinking water supplies (U.S. EPA, 1978a). The population most exposed to 1,1-DCE consists of workers in industries manufacturing or using 1,1-DCE (Hushon and Kornreich, 1976; Arthur D. Little, Inc., 1976). 1,1-DCE was identified as a cocontaminant with vinyl chloride monomer in the working environment of polyvinylchloride productions plants (Kramer and Mutchler, 1972).

Under laboratory conditions, the dichloroethylenes have shown to be toxic to fish (Dawson, et al. 1977; U.S. EPA, 1978b). The primary effect of acute toxicity of the dichloroethylenes is depression of the central nervous system. There is concern over the toxicity of 1,1-DCE because it is toxic to the liver of experimental animals (Jaeger, et al. 1975) and has a structure similar to vinyl chloride monomer, a known liver carcinogen in several species including humans (Maltoni, 1976; Spirtas and Kaminski, 1978).

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AQUATIC LIFE TOXICOLOGY*

FRESHWATER ORGANISMS

Introduction

All of the available data for dichloroethylenes with one exception are for 1,1-dichloroethylene. The bluegill has been tested (U.S. EPA, 1978) with both 1,1-dichloroethylene and 1,2-dichloroethylene under similar conditions, and the 96-hour LC50 values under static conditions were 73,900 and 135,000 $\mu\text{g/l}$, respectively. Apparently, the location of the chlorine atoms on the molecule does not affect the acute toxicity of dichloroethylenes very much.

Acute Toxicity

Most of these data have been determined using static test procedures. Dill, et al. (manuscript) tested the fathead minnow and they observed that the 96-hour LC50 was 169,000 $\mu\text{g/l}$ using static techniques and 108,000 $\mu\text{g/l}$ using flow-through techniques with measured concentrations. The value from the static test gave a lower estimate of toxicity than the flow-through test. The only

*The reader is referred to the Guidelines for Deriving Water Quality Criteria for the Protection of Aquatic Life [43 FR 21506 (May 18, 1978) and 43 FR 29028 (July 5, 1978)] and the Methodology Document in order to better understand the following discussion and recommendation. The following tables contain the appropriate data that were found in the literature, and at the bottom of each table are the calculations for deriving various measures of toxicity as described in the Guidelines.

additional data are for the bluegill; some of these results were discussed above. This species is about as sensitive to 1,1-dichloroethylene as the fathead minnow. The Final Fish Acute Values for 1,1-dichloroethylene and 1,2-dichloroethylene are 21,000 and 19,000 $\mu\text{g}/\text{l}$, respectively (Table 1).

The data on acute static tests with bluegill under similar conditions (U.S. EPA, 1978) in this and other documents on structurally related chemicals show a correlation between degree of chlorination and toxicity. The 96-hour LC50 values for this species are 73,900 and 135,000 $\mu\text{g}/\text{l}$ for 1,1- and 1,2-dichloroethylene, respectively, and 44,700 $\mu\text{g}/\text{l}$ for trichloroethylene and 12,900 $\mu\text{g}/\text{l}$ for tetrachloroethylene. These results indicate an increase in the lethal effect on bluegills with an increase in chlorine content. The correlation with Daphnia magna toxicity is not as clear. The 48-hour LC50 values are 79,000, 85,200, and 17,700 $\mu\text{g}/\text{l}$ for 1,1-dichloroethylene, trichloroethylene, and tetrachloroethylene, respectively.

Two 48-hour tests were conducted with Daphnia magna with 1,1-dichloroethylene and the EC50 values were 11,600 $\mu\text{g}/\text{l}$ (Dill, et al. manuscript) and 79,000 $\mu\text{g}/\text{l}$ (U.S. EPA, 1976). The cause of this difference cannot be ascertained. The resultant Final Invertebrate Acute Value for this chemical is 1,200 $\mu\text{g}/\text{l}$ (Table 2) which becomes the Final Acute Value since it is lower than the equivalent value for fish.

Chronic Toxicity

An embryo-larval test has been conducted (U.S. EPA, 1978) with the fathead minnow and 1,1-dichloroethylene. The range of experimental concentrations was such that no adverse effects were observed at the highest test concentration of 2,800 µg/l. The flow-through 96-hour LC50 for this species was 108,000 µg/l (Dill, et al. manuscript) suggesting that the range of concentration between acute and chronic effects for the fathead minnow is less than 40 times. The Final Fish Chronic Value is greater than 210 µg/l which is derived by dividing the chronic value by the sensitivity factor of 6.7 (Table 3). Invertebrate or other fish species have not been tested for chronic effects.

Plant Effects

The 96-hour EC50 values (Table 4), based on chlorophyll a and cell numbers of the alga Selenastrum capricornutum, were greater than 798,000 µg/l (U.S. EPA, 1978).

Miscellaneous

Dill, et al. (manuscript) extended their testing with the fathead minnow and observed a 13-day LC50 of 29,000 µg/l for 1,1-dichloroethylene (Table 5). Their equivalent 96-hour LC50 value (Table 1) was 108,000 µg/l which indicates that significant mortality occurred between 96 hours and 13 days.

CRITERION FORMULATION

Freshwater - Aquatic Life

Summary of Available Data

The concentrations below have been rounded to two significant figures.

1,1-dichloroethylene

Final Fish Acute Value = 16,000 $\mu\text{g/l}$

Final Invertebrate Acute Value = 1,200 $\mu\text{g/l}$

Final Acute Value = 1,200 $\mu\text{g/l}$

Final Fish Chronic Value = greater than 210 $\mu\text{g/l}$

Final Invertebrate Chronic Value = not available

Final Plant Value = greater than 800,000 $\mu\text{g/l}$

Residue Limited Toxicant Concentration = not available

Final Chronic Value = greater than 210 $\mu\text{g/l}$

$0.44 \times \text{Final Acute Value} = 530 \mu\text{g/l}$

1,2-dichloroethylene

Final Fish Acute Value = 19,000 $\mu\text{g/l}$

Final Invertebrate Acute Value = not available

Final Acute Value = 19,000 $\mu\text{g/l}$

Final Fish Chronic Value = not available

Final Invertebrate Chronic Value = not available

Final Plant Value = not available

Residue Limited Toxicant Concentration = not available

Final Chronic Value = not available

$0.44 \times \text{Final Acute Value} = 8,400 \mu\text{g/l}$

No freshwater criterion can be derived for either 1,1-dichloroethylene or 1,2-dichloroethylene using the Guidelines

because no Final Chronic Value for either fish or invertebrate species or a good substitute for either value is available.

However, results obtained for structurally similar compounds provide a way to estimate a criterion. For tetrachloroethylene and saltwater organisms and for trichloroethylene and freshwater organisms the Final Chronic Value is greater than 0.44 times the Final Acute Value, even though chronic values are available for the mysid shrimp and Daphnia magna, respectively. Therefore, this relationship is assumed in order to estimate criteria for both dichloroethylenes and freshwater organisms and 0.44 times the Final Acute Value is used.

Both a Final Fish Acute Value and a Final Invertebrate Acute Value are available for 1,1-dichloroethylene, and the Final Acute Value is based on the lower invertebrate value. Although a Final Fish Acute Value is available for 1,2-dichloroethylene, no invertebrate data are available. In all cases for di-, tri- and tetrachloroethylenes in which both Final Fish and Final Invertebrate Acute Values are available, the invertebrate species value is lower. Therefore the missing acute data for invertebrates and 1,2-dichloroethylene is important. In deriving an estimated criterion, the relative sensitivity of fish and invertebrate species for 1,1-dichloroethylene is assumed to hold for 1,2-dichloroethylene as well. For 1,1-dichloroethylene the Final Invertebrate Acute Value divided by the Final Fish Acute Value is $1,200/16,000 = 0.075$. Multiplying this value times the Final Fish Acute Value for 1,2-dichloroethylene results in an estimated Final Invertebrate Acute Value of $0.075 \times 19,000 \mu\text{g/l} = 1,400 \mu\text{g/l}$.

Thus the Final Acute Value for 1,2-dichloroethylene is 1,400 µg/l. Multiplying the Final Acute Value of 1,400 µg/l by 0.44 gives 620 µg/l.

1,1-dichloroethylene

The maximum concentration of 1,1-dichloroethylene is the Final Acute Value of 1,200 µg/l and the 24-hour average concentration is 0.44 times the Final Acute Value. No important adverse effects on freshwater aquatic organisms have been reported to be caused by concentrations lower than the 24-hour average concentration.

CRITERION: For 1,1-dichloroethylene the criterion to protect freshwater aquatic life as derived using procedures other than the Guidelines is 530 µg/l as a 24-hour average and the concentration should not exceed 1,200 µg/l at any time.

1,2-dichloroethylene

The maximum concentration of 1,2-dichloroethylene is the estimated Final Acute Value of 1,400 µg/l and the 24-hour average concentration is 0.44 times the Final Acute Value. No important adverse effects on freshwater aquatic organisms have been reported to be caused by concentrations lower than the 24-hour average concentration.

CRITERION: For 1,2-dichloroethylene the criterion to protect freshwater aquatic life as derived using procedures other than the Guidelines is 620 µg/l as a 24-hour average and the concentration should not exceed 1,400 µg/l at any time.

Table 1. Freshwater fish acute values for dichloroethylenes

Organism	Bioassay Method	Test Conc., **	Chemical Description	Time (hrs)	LC50 (ug/l)	Adjusted LC50 (ug/l)	Reference
Fathead minnow, <u>Pimephales promelas</u>	S	U	1,1-dichloroethylene	96	169,000	92,000	Dill, et al. Manuscript
Fathead minnow, <u>Pimephales promelas</u>	FT	M	1,1-dichloroethylene	96	108,000	108,000	Dill, et al. Manuscript
Bluegill, <u>Lepomis macrochirus</u>	S	U	1,1-dichloroethylene	96	73,900	40,401	U.S. EPA, 1978
Bluegill, <u>Lepomis macrochirus</u>	S	U	1,2-dichloroethylene	96	135,000	73,805	U.S. EPA, 1978

* S = static, FT = flow-through

** U = unmeasured, M = measured

Geometric mean of adjusted values: 1,1-dichloroethylene = $63,460 \text{ ug/l}$

$$\frac{63,460}{3.9} = 16,000 \text{ ug/l}$$

1,2-dichloroethylene = $73,805 \text{ ug/l}$

$$\frac{73,805}{3.9} = 19,000 \text{ ug/l}$$

Lowest value from a flow-through test with measured concentrations = $108,000 \text{ ug/l}$ for 1,1-dichloroethylene

Table 2. Freshwater invertebrate acute values for dichloroethylenes

Organism	Bioassay Method	Test Conc.**	Chemical Description	Time (hrs)	LC50 (ug/l)	Adjusted LC50 (ug/l)	Reference
Cladoceran, <u>Daphnia magna</u>	S	U	1,1-dichloroethylene	48	11,600	9,800	Dill, et al. Manuscript
Cladoceran, <u>Daphnia magna</u>	S	U	1,1-dichloroethylene	48	79,000	66,913	U.S. EPA, 1978

* S = static

** U = unmeasured

Geometric mean of adjusted values = $25,608 \text{ ug/l}$ $\frac{25,608}{2} = 1,200 \text{ ug/l}$

Table 3. Freshwater fish chronic values for dichloroethylenes (U.S. EPA, 1978)

Organism	Test*	Limits** (ug/l)	Chronic Value (ug/l)
<u>1,1-dichloroethylene</u>			
Fathead minnow, <u>Pimephales promelas</u>	E-L	>2,800	>1,400

* E-L = embryo-larval

Geometric mean for chronic values = $\frac{1,400}{6.7} \mu\text{g/l}$

$\frac{1,400}{6.7} = >210 \mu\text{g/l}$

Lowest chronic value = $>1,400 \mu\text{g/l}$

Table 4. Freshwater plant effects for dichloroethylenes (U.S. EPA, 1978)

<u>Organism</u>	<u>Effect</u>	<u>Concentration</u> <u>(ug/l)</u>
<u>Alga.</u> <u>Selenastrum</u> <u>capricornutum</u>	EC50 96-hr chlorophyll <u>a</u>	<u>1,1-dichloroethylene</u> >798,000
<u>Alga.</u> <u>Selenastrum</u> <u>capricornutum</u>	EC50 96-hr cell count	>798,000

Lowest plant value = >798,000 ug/l

Table 5. Other freshwater data for dichloroethylenes (Dill, et al. Manuscript)

<u>Organism</u>	<u>Test</u> <u>Duration</u>	<u>Effect</u>	<u>Result</u> <u>µg/L</u>
Fathead minnow, <u>Pimephales promelas</u>	<u>13 days</u>	<u>1,1-dichloroethylene</u> LC50	29,000

SALTWATER ORGANISMS

Introduction

All of the data on the effects of dichloroethylenes on saltwater organisms are for 1,1-dichloroethylene.

Acute Toxicity

The unadjusted 96-hour LC50 values (Table 6) for the sheeps-head minnow (U.S. EPA, 1978) and the tidewater silversides (Dawson, et al. 1977) are 249,000 and 250,000 $\mu\text{g/l}$, respectively. The 96-hour LC50 for mysid shrimp (Table 7) is 224,000 $\mu\text{g/l}$ which result is similar to that for the tested fish species. However, since the species sensitivity factor for invertebrate species is much greater than that for fish, the final values for saltwater fish and invertebrate species are 37,000 and 3,900 $\mu\text{g/l}$, respectively, and the lower of the two becomes the Final Acute Value for 1,1-dichloroethylene.

The 96-hour LC50 for the mysid shrimp and 1,1-dichloroethylene is, as stated above, 224,000 $\mu\text{g/l}$ (U.S. EPA, 1978). The 96-hour LC50 for the same species under similar test conditions (U.S. EPA, 1978) is 10,200 $\mu\text{g/l}$ for tetrachloroethylene. As was suggested in the freshwater part of this document, acute toxicity of these structurally related compounds increases with increases in degree of chlorination.

Chronic Toxicity

No chronic toxicity data are available for saltwater species and any dichloroethylene.

Plant Effects

There was no effect of 1,1-dichloroethylene on the alga Skeletonema costatum at test concentrations as high as 712,000 $\mu\text{g/l}$ (Table 8).

CRITERION FORMULATION

Saltwater - Aquatic Life

Summary of Available Data

The concentrations below have been rounded to two significant figures.

1,1-dichloroethylene

Final Fish Acute Value = 37,000 µg/l

Final Invertebrate Acute Value = 3,900 µg/l

Final Acute Value = 3,900 µg/l

Final Fish Chronic Value = not available

Final Invertebrate Chronic Value = not available

Final Plant Value = greater than 710,000 µg/l

Residue Limited Toxicant Concentration = not available

Final Chronic Value = greater than 710,000 µg/l

0.44 x Final Acute Value = 1,700 µg/l

1,2-dichloroethylene

Final Fish Acute Value = not available

Final Invertebrate Acute Value = not available

Final Acute Value = not available

Final Fish Chronic Value = not available

Final Invertebrate Chronic Value = not available

Final Plant Value = not available

Residue Limited Toxicant Concentration = not available

Final Chronic Value = not available

0.44 x Final Acute Value = not available

No saltwater criterion can be derived for either 1,1- or 1,2-dichloroethylene using the Guidelines because no Final Chronic Value for either fish or invertebrate species or a good substitute for either value is available.

However, results obtained for structurally similar compounds indicate how a criterion may be estimated. For tetrachloroethylene and saltwater organisms and for trichloroethylene and freshwater organisms the Final Chronic Value is greater than 0.44 times the Final Acute Value, even though chronic values are available for the mysid shrimp and Daphnia magna, respectively. Therefore, this relationship is assumed in order to estimate a criterion for dichloroethylenes and saltwater organisms and 0.44 times the Final Acute Value is used.

Both a Final Fish Acute Value and a Final Invertebrate Acute Value are available for 1,1-dichloroethylene, and the Final Acute Value is equal to the lower invertebrate value. Because neither a Final Fish Acute Value nor a Final Invertebrate Acute Value is available for 1,2-dichloroethylene, no criterion can be derived.

1,1-dichloroethylene

The maximum concentration of 1,1-dichloroethylene is the Final Acute Value of 3,900 µg/l and the 24-hour average concentration is 0.44 times the Final Acute Value. No important adverse effects on saltwater aquatic organisms have been reported to be caused by concentrations lower than the 24-hour average concentration.

CRITERION: For 1,1-dichloroethylene the criterion to protect saltwater aquatic life as derived using procedures other than the Guidelines is 1,700 $\mu\text{g}/\text{l}$ as a 24-hour average and the concentration should not exceed 3,900 $\mu\text{g}/\text{l}$ at any time.

1,2-dichloroethylene

No saltwater criterion can be derived for 1,2-dichloroethylene using the Guidelines because no Final Chronic Value for either fish or invertebrate species or a good substitute for either value is available, and there are insufficient data to estimate a criterion using other procedures.

Table 6. Marine fish acute values for dichloroethylenes

Organism	Bioassay Method*	Test Conc.**	Time (hrs)	LC50 (ug/l)	Adjusted LC50 (ug/l)	Reference
1,1-Dichloroethylene						
Sheepshead minnow, <u>Cyprinodon variegatus</u>	S	U	96	249,000	136,128	U.S. EPA, 1978
Tidewater silversides, <u>Menidia beryllina</u>	S	U	96	250,000	136,675	Dawson, et al., 1977

* S = static

** U = unmeasured

Geometric mean of adjusted values = $136,128 \mu\text{g/l}$ $\frac{136,128}{3.4} = 37,000 \mu\text{g/l}$

Table 7. Marine invertebrate acute values for dichloroethylenes (U.S. EPA, 1978)

Organism	Bioassay Test Method	Time (hrs)	LC50 (ug/l)	Adjusted LC50 (ug/l)
Mysid shrimp, <u>Mysidopsis bahia</u>	S U	96	224,000	189,728

* S = static

** U = unmeasured

Geometric mean of adjusted values = 189,728 ug/l

$$\frac{189,728}{49} = 3,900 \text{ ug/l}$$

Table 8. Marine plant effects for dichloroethylenes (U.S. EPA, 1978)

<u>Organism</u>	<u>Effect</u>	<u>Concentration</u> <u>(ug/l)</u>
		1,1-Dichloroethylene
Alga.	No effect	>712,000
<u>Skeletonema costatum</u>		

Lowest plant value = >712,000 ug/l

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U.S. EPA. 1978. In-depth studies on health and environmental impacts of selected water pollutants. Contract No. 68-01-4646. U.S. Environ. Prot. Agency.

Mammalian Toxicology and Human Health Effects

Exposure

Ingestion in Water. The National Organic Monitoring Survey (U.S. EPA, 1978a) reported detecting, but did not quantify occurrence of one of the three dichloroethylenes (DCE) in finished drinking water, 1,1-dichloroethylene (vinylidene chloride). Fishbein (1976) indicates there is no information concerning the amount of 1,1-DCE which may migrate from discarded materials made from vinylidene chloride polymers into water. Another source of 1,1-DCE is from the decomposition of 1,1,1-trichloroethylene (McConnell, et al. 1975) which has been occasionally detected in drinking water (U.S. EPA, 1978b) usually at low concentrations (about 1.0 ug/l). The other dichloroethylenes cis-1,2-dichloroethylene (cis-1,2-DCE) and trans-1,2-dichloroethylene (trans-1,2-DCE) were found at concentrations of 16 and 1 ug/l in Miami drinking water (U.S. EPA, 1975, 1978b). Concentrations of 0.1 ug/l cis-1,2-DCE were observed in Cincinnati and Philadelphia drinking waters as well, but were absent from seven other drinking waters included in this survey. Much too little information is available to estimate and assess current exposure of DCE to humans via drinking water.

Ingestion in Food. Food wrappers are commonly made of 1,1-DCE copolymers. However, there appear to be no data as to the extent to which the untreated monomer migrates into foods (Fishbein, 1976). The DCE's are very volatile compounds and

are degraded in air with a half-life of 8 weeks (Pearson, et al. 1975).

A bioconcentration factor (BCF) relates the concentration of a chemical in water to the concentration in aquatic organisms, but BCF's are not available for the edible portions of all four major groups of aquatic organisms consumed in the United States. Since data indicate that the BCF for lipid-soluble compounds is proportional to percent lipids, BCF's can be adjusted to edible portions using data on percent lipids and the amounts of various species consumed by Americans. A recent survey on fish and shellfish consumption in the United States (Cordle, et al. 1978) found that the per capita consumption is 18.7 g/day. From the data on the 19 major species identified in the survey and data on the fat content of the edible portion of these species (Sidwell, et al. 1974), the relative consumption of the four major groups and the weighted average percent lipids for each group can be calculated:

<u>Group</u>	<u>Consumption (Percent)</u>	<u>Weighted Average Percent Lipids</u>
Freshwater fishes	12	4.8
Saltwater fishes	61	2.3
Saltwater molluscs	9	1.2
Saltwater decapods	18	1.2

Using the percentages for consumption and lipids for each of these groups, the weighted average percent lipids is 2.3 for consumed fish and shellfish.

No measured steady-state bioconcentration factor (BCF) is available for 1,1-dichloroethylene, but the equation " $\text{Log BCF} = 0.76 \text{ Log P} - 0.23$ " can be used (Veith, et al. Manuscript) to estimate the BCF for aquatic organisms that contain about 8 percent lipids from the octanol-water partition coefficient of 135, the steady-state bioconcentration factor for 1,1-dichloroethylene is estimated to be 24. An adjustment factor of $2.3/8.0 = 0.2875$ can be used to adjust the estimated BCF from the 8.0 percent lipids on which the equation is based to the 2.3 percent lipids that is the weighted average for consumed fish and shellfish. Thus, the weighted average bioconcentration factor for 1,1-dichloroethylene and the edible portion of all aquatic organisms consumed by Americans is calculated to be $24 \times 0.2875 = 6.9$.

Dermal. There are no data indicating the extent and significance of human exposure to dichloroethylenes via the skin.

Pharmacokinetics

Absorption. No animal or human studies appear to exist which specifically document the degree of systemic absorption of the DCE's by any route. However, related compounds such as trichloroethylene and tetrachloroethylene are absorbed to near the theoretical maximum for both the inhalation and ingestion routes (Daniel, 1963; Munster, et al. 1976). On this basis 35 to 50 percent of inhaled DCE and virtually 100 percent of ingested DCE may be absorbed systemically. Such a high degree of absorption of 1,1-DCE seems implied by the studies of McKenna, et al. (1977a,b) even if not specifically quantified.

Distribution. Distribution of 1,1-DCE in various organs of the rat has been studied after inhalation exposure (Jaeger, et al. 1977a) using ¹⁴C-labeled 1,1-DCE. The distribution of total radioactivity 30 minutes after cessation of exposure was similar in fed and fasted rats, although the absolute level of label was considerably greater in fasted animals. The largest concentrations were found in kidney, followed by liver, spleen, heart, and brain. Blood concentrations were high relative to tissue concentrations. A similar distribution was observed for trichloroacetic acid-insoluble (TCA-insoluble), labeled material (presumably bound to macromolecules).

Subcellular distribution of radio-labeled 1,1-DCE metabolites. 30 minutes following a 2-hour exposure to 8,000 $\mu\text{g}/\text{m}^3$ was observed in the microsomal - mitochondrial cytosolic compartments of the liver (Jaeger, et al. 1977a). However, a large amount of radioactivity in the cytosol was TCA- and CHCl_3 soluble, whereas that in the mitochondria and microsomes was insoluble. These data suggest substantial binding of 1,1-DCE metabolites to macromolecules and association with the lipid present in the latter two fractions, respectively. The turnover of bound TCA-insoluble radioactivity derived from 1,1-DCE has a half-life of 2 to 3 hours.

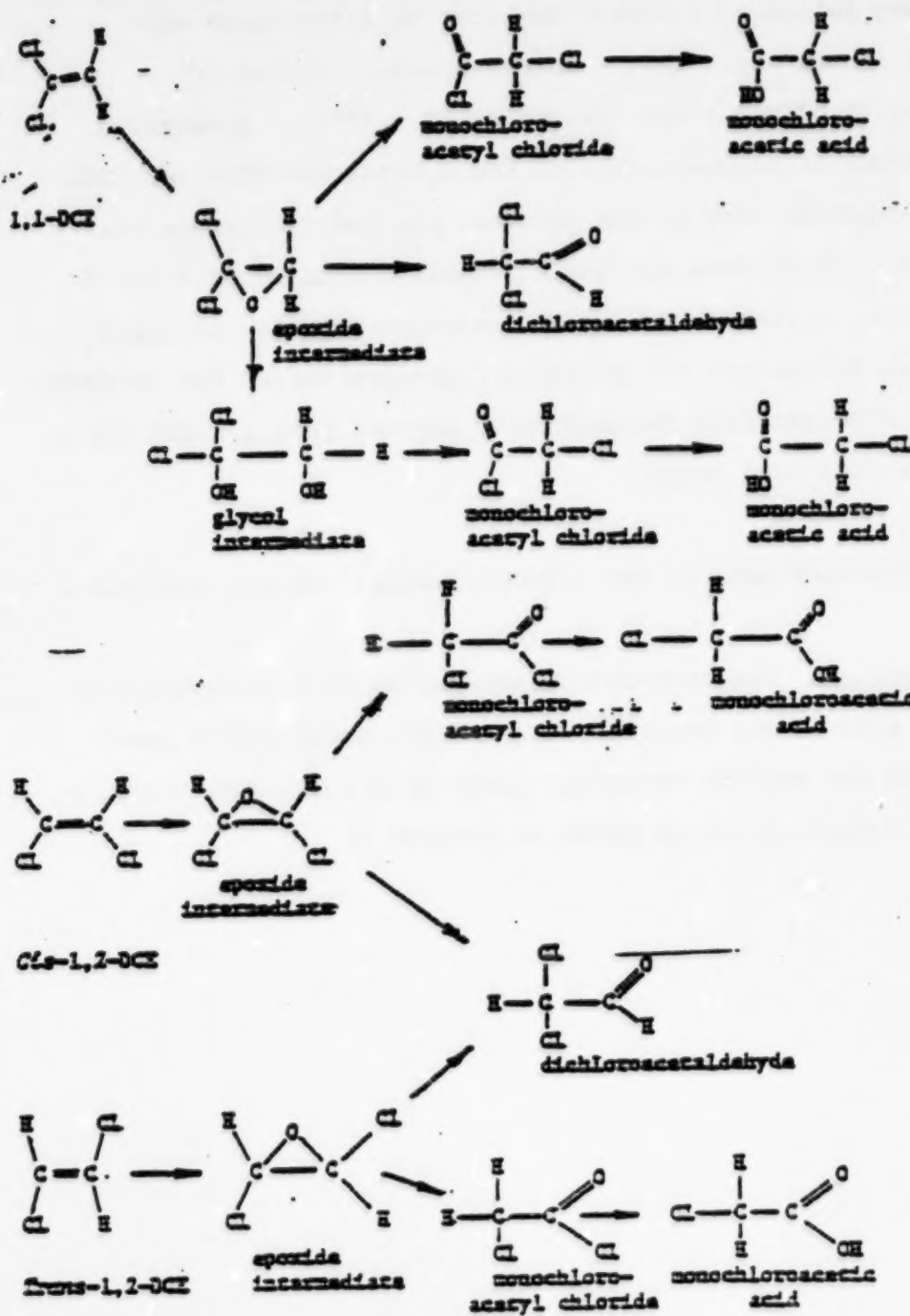
Distribution data on the 1,2-DCE isomers are not available.

Metabolism. The comparative metabolism of chloroethylenes has been extensively studied. Liebman and Ortiz (1977) have postulated the various metabolic pathways for 1,1-DCE, cis-1,2-DCE, and trans-1,2-DCE as shown in Diagram 1.

Diagram 1. - Proposed metabolic pathways of 1,1-DCE and the 1,2-DCE isomers (Lieberman and Ortiz, 1977).

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In addition, there is evidence that the 1,1-DCE metabolites are conjugated with glutathione, which presumably represents a detoxification step (McKenna, et al. 1977a). Bonse, et al. (1975) identified both the chloroethanol and chloroacetic acid derivatives of cis-1,2-DCE and trans-1,2-DCE in a perfused rat liver. The metabolism of the cis-isomer relative to the amount taken up by the liver was much greater than the trans-isomer. Liebman and Ortiz (1977) identified formation of chloroacetic acid from 1,1-DCE, but as yet the dichloroacetaldehyde has not been unequivocally identified as a metabolite of 1,1-DCE. Inhibition of epoxide hydrolase resulted in a stimulation of chloroacetic acid formation from 1,1-DCE, leading to the conclusion that the glycol intermediate is relatively unimportant in the conversion of 1,1-DCE to chloroacetic acid (Liebman and Ortiz, 1977). The essential feature of each of these metabolic pathways is that all chloroethylenes appear to be metabolized through epoxide intermediates which are reactive and may form covalent bonds with tissue macromolecules (Henschler, 1977a).

In the intact animal, a large portion of the systemically absorbed 1,1-DCE is metabolically converted. Jaeger, et al. (1977a) found 36.7 ± 3.3 percent of the absorbed dose in the urine of rats within 26 hours. Disposition of ¹⁴C-activity from radiolabeled 1,1-DCE in mice and rats over a 48-hour period (McKenna, et al. 1977b) is shown in Table 10.

Table 10. - Disposition of 1,1-DCE following
inhalation of 40 mg/m³ over a
48-hour period (McKenna, 1977b)

	Mice	Rats
Expired 1,1-DCE %	0.65 ± 0.07	1.63 ± 0.14
Expired CO	4.64 ± 0.17	3.74 ± 3.72
Body Burden %		
Urine	80.83 ± 1.58	74.72 ± 2.30
Feces	6.58 ± 0.81	9.73 ± 0.10
Carcass	5.46 ± 0.41	4.75 ± 0.78
Cage Wash.	1.83 ± 0.84	0.44 ± 0.28
Body Burden mg.eq.		
1,1-DCE/kg	5.30 ± 0.75	2.89 ± 0.24
Total metabolized		
mg.eq. 1,1-DCE/kg	5.27 ± 0.74	2.84 ± 0.26

It is clear from these data that the mouse develops a higher body burden of 1,1-DCE than the rat at identical exposure levels. The disposition of 1,1-DCE appears quite similar in the two species. However, as a result of the overall greater rate of metabolism covalently bound 1,1-DCE metabolites are higher in the mouse than in the rat as shown in Table 2. The substantial differences in distribution may be related to the differing

sensitivity of the species to carcinogenic effects of 1,1-DCE ..
(Bathaway, 1977).

Table 11. - Covalently bound ^{14}C -activity in rat
and mouse following exposure to 40 mg/m^3
 ^{14}C -1,1-DCE (McKenna, et al. 1977b)

	<u>^{14}C-1,1-DCE, $\mu\text{g.eq/mg protein}$</u>	
	<u>Liver</u>	<u>Kidney</u>
Mice	22.29 ± 3.77	79.55 ± 19.11
Rats .	5.28 ± 0.14	13.14 ± 1.15

It is notable that in both species covalently bound metabolites of 1,1-DCE are highest in kidney followed by liver. Exposure of rats to a higher concentration (8000 mg/m^3) for a shorter period (2 hours) appears to give somewhat differing results. In this case a higher degree of TCA-insoluble ^{14}C -activity is observed in liver. These differences between long- and short-term exposures must be kept in mind when evaluating the long-term toxicity of 1,1-DCE.

The relationship of metabolism of DCE's to their toxicity is not well understood. Differing results are obtained with inducers and inhibitors of microsomal enzymes depending upon the age or body weight of experimental animals (Anderson and Jenkins, 1977). Pretreatment with dithiocarbamates which presumably

interfere with 1,1-DCE metabolism, protects against lethal and hepatotoxic damage (from 1,1-DCE) and reduces covalent binding of metabolites to tissue macromolecules (Short, et al. 1977a). On the other hand, Aroclor 1254 or phenobarbital pretreatment, which would increase microsomal enzyme activity, also decreases the hepatotoxicity of 1,1-DCE (Reynolds, et al. 1975; Jenkins, et al. 1972). However, Carlson and Fuller (1972) reported increased mortality from 1,1-DCE in rats following phenobarbital pretreatment. The possibility exists that these pretreatments interact with more than one metabolic pathway and/or differing mechanisms account for hepatotoxicity and lethality of 1,1-DCE and this accounts for the inconsistent responses (Anderson and Jenkins, 1977; Reynolds and Moslen, 1977). It is clear that the hepatotoxicity of 1,1-DCE does increase with decreasing concentrations of hepatic glutathione (Jaeger, et al. 1973). This seems involved with the greater sensitivity of fasted animals to 1,1-DCE-induced hepatotoxicity (Jaeger, et al. 1974). In regard to this issue, Hathaway (1977) has identified thioglycollic acid, thioglycollic acid, dithioglycollic, and an N-acetyl-S-cysteinyl-acetyl derivative as products of 1,1-DCE metabolism in rats and mice.

Excretion. There appear to be no data relating to the rate at which any of the DCE's are cleared from the body. In the case of 1,1-DCE one can guess that the rate of elimination is relatively rapid in that substantial fractions of the total absorbed dose may be recovered in urine within 26 or 72 hours (Jaeger, et al. 1977a; McKenna, et al. 1977a). However, these

data do not allow a more precise estimate of turnover.

Disappearance of covalently bound metabolites of 1,1-DCE (measured as TCA-insoluble fractions) appears to be fairly rapid as well, with a reported half-life of 2 to 3 hours (Jaeger, et al. 1977a). These data were obtained from experiments with insufficient duration and resolution of the nature of the binding to rule out the possibility of a residual binding to tissue macromolecules of an irreversible or slowly reversible character.

No data exist concerning the excretion of the cis- or trans-isomers of 1,2-DCE.

Effects

Toxicity. Like other members of the chlorinated ethylene series, the DCE's possess anesthetic properties. Unlike those compounds of the group which were employed as general anesthetics, very little effort has gone into characterizing either short- or long-term CNS toxicity of the DCE's. Irish (1962) referenced unpublished data concerning the CNS depressant activity of 1,1-DCE in humans. A concentration of 16,000 mg/m^3 was estimated as sufficient to rapidly produce a state resembling drunkenness and was felt likely to result in unconsciousness if exposure was continued. A single report has been made associating sensory disturbances in the trigeminal area of the face of a person following accidental exposure to 1,1-DCE (Brinser, et al. 1970). Subsequent evaluation, however, suggested that the toxic agent was either mono- or di-chloroacetylene which

was a contaminant of 1,1-DCE (Benschler, et al. 1970). No further information exists concerning the central nervous system toxicity of these compounds.

In recent years, considerable attention has focused on the liver and kidney damage produced by 1,1-DCE. Prendergast, et al. (1967) documented definite morphological damage to kidney and liver of rats and guinea pigs exposed to 189 mg 1,1-DCE/m³. These effects were associated with an increased liver lipid concentration in rats. In guinea pigs, continuous 90-day exposure to concentrations as low as 20 mg/m³ produced increased mortality, whereas intermittent exposures (30 exposures, 8 hr/day, 5 days/week) at 395 mg/m³ produced no increase in mortality. A similar difference in intermittent versus continuous exposures was observed in the monkey where increased mortality was observed with 101 mg/m³ of continuous exposure and no mortality observed with the same intermittent exposure utilized for guinea pigs.

Results of a 90-day toxicity study in rats given 1,1-DCE in water or in air revealed cytoplasmic vacuolization in liver cells of rats ingesting 200 ppm (0.79 mg/l) or inhaling 25 ppm (6 mg/m³) of the compound. The hepatocellular changes were interpreted to be of a reversible character (Quast, et al. 1977).

Oral administration of single doses of 200 and 400 mg 1,1-DCE/kg decreases liver glucose-6-phosphatase, increases liver alkaline phosphatase, liver tyrosine transaminase, plasma

alkaline phosphatase, and plasma alanine transaminase (Jenkins, et al. 1972) 20 hours after administration. The LD50 of 1,1-DCE was decreased in adrenalectomized rats (1,550 mg/kg in sham-operated, 34 mg/kg in adrenalectomized animals) by a factor of 20. This effect was not clearly related to the hepatotoxicity of 1,1-DCE (Jenkins, et al. 1972).

The hepatotoxic effects of 1,1-DCE appear to be potentiated by depletion of hepatic glutathione concentrations, whether resulting from normal diurnal variation (Jaeger, et al. 1973) or by fasting (Jaeger, et al. 1974, 1975). In the latter case it was possible to demonstrate increased concentrations of 1,1-DCE in blood and liver in the fasted animals. Thyroidectomy decreases whereas thyroxine administration exacerbated the hepatotoxic effects of 1,1-DCE (Szabo, et al. 1977; Jaeger, et al. 1977b). At the same time thyroidectomy increased, and thyroxine decreased liver glutathione concentrations. Jaeger (1977) has suggested that the hepatotoxic effects of 1,1-DCE are secondary to reduction of mitochondrial glutathione concentrations (and sulfhydryl enzymes) and marked inhibition of mitochondrial respiration.

Jenkins, et al. (1972) found both cis- and trans-1,2-DCE to be considerably less potent than 1,1-DCE as a hepatotoxin. Freundt, et al. (1977) indicated that repeated inhalation exposures of 800 mg/m³ (8 hours/day, 5 days/week, 16 weeks) of the trans-1,2-DCE produces fatty degeneration of the liver.

Less attention has been paid to the renal toxicity of the DCE's despite the occurrence of histologically demonstrated damage at 1,1-DCE exposures equal to or less than those required for hepatotoxicity (Prendergast, et al. 1967; Short, et al. 1977a). The degree of covalent binding of 1,1-DCE metabolites has been shown to be higher in kidney than in liver in both rats and mice (McKenna, et al. 1977b; Short, et al. 1977a) suggesting that renal damage resulting from 1,1-DCE should be more carefully examined in the future. No quantitative data appear available on the nephrotoxicity of 1,2-DCE's.

Inhalation of 1,1-DCE at high concentrations ($102,000 \text{ mg/m}^3$) for short periods of time (10 minutes) has been found to sensitize the myocardium to arrhythmias produced by injection of epinephrine (Silatchnik and Carlson, 1974). Unlike hepatotoxicity, the cardiac sensitizing effects of 1,1-DCE were enhanced by phenobarbital pretreatment. This implies that a metabolite of 1,1-DCE may be involved. Similar effects have been observed in human poisoning with trichloroethylene. The 1,2-DCE isomers have not been investigated with respect to this effect.

Only one epidemiological study has been published which examined workers exposed to 1,1-DCE uncomplicated by exposures to other solvents (Ott, et al. 1976). At the time of this preliminary report, no abnormal findings could be associated with 1,1-DCE exposure in a population of 138 workers. Measured concentrations in the workplaces of these individuals ranged from 9 to 280 mg/m^3 (time-weighted averages).

As reported with a number of other chlorinated hydrocarbon solvents, 1,1-DCE increases bile-duct pancreatic fluid flow in fasted rats (Hamada and Peterson, 1977). The composition of the fluid did not differ significantly from that in control animals. The significance of this finding remains to be established.

Synergism and/or Antagonism. DCEs are metabolically converted to reactive epoxide intermediates (Bonse, et al. 1975; Hathaway, 1977; McKenna, et al. 1977b). Consequently, it would be predicted that compounds which increase or decrease the rate of DCE metabolism would affect toxicity. However, interactions at this level do not at present lend themselves to simple prediction. Compounds which decrease covalent binding of 1,1-DCE metabolites, such as disulfiram, protect against lethality and hepatotoxicity resulting from 1,1-DCE exposure (Short, et al. 1977a). On the other hand, pretreatment of animals with inducers of microsomal enzyme systems, such as Aroclor 1254 or phenobarbital, appear to decrease the hepatotoxicity due to 1,1-DCE (Reynolds, et al. 1975; Jenkins, et al. 1972), but increase mortality (Carlson and Fuller, 1972). These conflicting findings suggest that metabolism of the DCEs is not sufficiently understood to allow straightforward predictions. It is likely that the degree of conjugation of reactive intermediates with glutathione may be the factor not yet taken into account in attempting to predict the impact of altering microsomal enzyme activities on toxicity (Hathaway, 1977).

Altering tissue glutathione concentrations affects the hepatotoxicity of 1,1-DCE (Jaeger, et al. 1973, 1977). Decreased tissue glutathione is associated with greater toxicity and elevated glutathione with decreased toxicity in response to 1,1-DCE challenge.

Thyroidectomy decreases and adrenalectomy greatly potentiates the toxicity of 1,1-DCE (Szabo, et al. 1977; Jenkins, et al. 1972), by altering glutathione levels.

Teratogenicity. Teratogenic effects of the DCEs do not appear to have been evaluated.

Mutagenicity. 1,1-DCE has been shown to be mutagenic in Salmonella typhimurium strains TA1530 and TA100 (Bartsch, et al. 1975) and E. coli K12 (Greim, et al. 1975). In both systems mutagenic activity required microsomal activation. Pretreatment with phenobarbital increased the mutagenic activity produced by microsomal fractions derived from liver, kidney or lung (Bartsch et al., 1975). Microsomal preparations, particularly liver, were considerably more active when derived from mice than from rats (Bartsch, et al., 1975).

Both the cis- and trans- isomers of 1,2-DCE were non-mutagenic when assayed with E. coli K12 at similar concentrations used for 1,1-DCE (Greim, et al. 1975). Henschler (1977a) and his associates have suggested that the mutagenic and presumably carcinogenic activities of the chloroethylene series are related

to the unsymmetrical chlorine substitution or the respective epoxide intermediates. Such substitution would result in less stable and more reactive intermediates than symmetrically substituted epoxides. These data support this hypothesis, at least with respect to mutagenesis in the E. Coli. K12 system. However, generalization of this hypothesis to carcinogenesis in intact animals is not yet possible and requires modification to account for the demonstrated carcinogenic activity of tetrachloroethylene (Nat'l. Cancer Inst., 1977). In addition, both 1,1-DCE and cis-1,2-DCE were found mutagenic in Salmonella tester strains (Cerna and Kypenova, 1977). The trans-1,2-DCE isomer was found inactive. Of the 3 DCE's only cis-1,2-DCE promoted chromosomal aberrations in cytogenic analysis of bone marrow cells with repeated i.p. injections (Cerna and Kypenova, 1977).

The finding of increased mutation rates in bacterial systems has not been confirmed in mammalian systems. Adult CD male rats exposed to 220 mg 1,1-DCE/m³ for 6 hr/day 5 days a week for 11 weeks failed to produce dominant lethal mutations (Short, et al. 1977b). Similar results have been reported by Anderson, et al. (1977) in dominant lethal studies in CD-1 mice.

Carcinogenicity. The carcinogenicity of 1,1-DCE is currently being evaluated in studies sponsored by the National Cancer Institute (1978). No results are yet available. Maltoni, et al. (1977) and Maltoni (1977) have reported preliminary results with inhalation exposures to 1,1-DCE. Exposure conditions were 4

4 to 5 days/week for 52 weeks to 100 mg 1,1-DCE/m³.
 Animals at the time of the report had been observed for a total
 of 98 weeks. At this concentration 25 of 300 Swiss mice had
 developed kidney adenocarcinomas. No kidney adenocarcinomas have
 been observed in 200 control animals. The majority of tumors
 were observed in male mice as shown below.

Table 12. - Kidney adenocarcinomas in animals exposed
 to 100 mg/m³ 1,1-DCE (Maltoni, 1977)

<u>1,1-DCE</u>	<u>Sex</u>	<u>Total Animals at Risk</u>	<u>No. Animals with adenocarcinoma</u>	<u>%</u>
None	Male	126	0	0
None	Female	158	0	0
100 mg/m	Male	98	16	16
100 mg/m	Female	112	1	0.9

In this same study, no kidney adenocarcinomas were observed in
 Sprague-Dawley rats at exposures up to 800 mg/m³, 1,1-DCE.

Maltoni, et al. (1977) also observed a significant increase
 in mammary adenocarcinomas in Swiss mice inhaling 100 mg 1,1-
 DCE/m³ and Sprague-Dawley rats exposed to 600 mg/m³ under the
 same conditions cited above. The mouse data are presented in
 Table 13 and the rat data in Table 14. Experiments in female
 Sprague-Dawley rats exposed to 20 mg 1,1-DCE by gavage 4 to 5

days a week for 52 weeks resulted in a 42 percent incidence of mammary tumors, whereas control animals had a 34 percent incidence. These latter data were not analyzed statistically.

Table 13. - Incidence of Mammary Adenocarcinomas in Female Mice
Receiving VDC via inhalation (Maltoni, et al. 1977)

	<u>0 mg/m³</u>	<u>40 mg/m³</u>	<u>100 mg/m³</u>
Number with tumors	2/158	1/26	7/112
Survivors at time of kidney adenocarcinomas			
Incidence	0.013	0.039	0.063
p-value		0.068	0.029

Table 14. - Incidence of mammary tumors in rats inhaling vinylidene chloride (VDC) (Maltoni, et al. 1977)

	Dose (mg/m ³)					
	<u>0</u>	<u>40</u>	<u>100</u>	<u>200</u>	<u>400</u>	<u>600</u>
Number with tumors	32/100	15/30	12/30	15/30	18/30	35/60
Number initially						
Incidence	0.32	0.50	0.40	0.50	0.60	0.58
p-value		0.21	0.45	0.21	0.044	0.007

Hamsters exposed to the same conditions as the Swiss mice failed to exhibit an increased tumor incidence (Maltoni, et al. 1977).

In another study, Lee, et al. (1977) observed a small increase in hepatic hemangiosarcomas in animals exposed to 220 mg/m³ 1,1-DCE, 4 hr/day 5 days/week for 7 to 12 months. Although kidney pathology was observed, no mention was made of kidney adenocarcinomas.

Rampy, et al. (1977) exposed male and female Sprague-Dawley rats to 200 mg 1,1-DCE/l in drinking water (2 years) or 100 and 300 mg/m³ 1,1-DCE by inhalation (6 hr/day, 5 days/week for 18 months). Based on total tumor count there was no evidence of increased tumors in animals treated with 1,1-DCE.

The only human data concerning possible carcinogenic effects of 1,1-DCE in man appeared in the epidemiological study of approximately 30 employees by Ott, et al. (1976). No associations of cancer deaths could be made with exposure to 1,1-DCE. The population was too small to evaluate the carcinogenicity of 1,1-DCE.

CRITERION FORMULATION

Existing Guidelines and Standards

Standards that have been established for the DCE's are applicable primarily to occupational exposures. The threshold limit values (TLV) for these compounds presently established by the American Conference of Governmental Industrial Hygienists (ACGIH) are 40 mg/m³ (1,1-DCE) and 790 mg/m³ (1,2-DCE). These values allow daily exposures of 286 mg 1,1-DCE per day and 5,643 mg 1,2-DCE per day. These calculations are based on the assumption of a 50 m³/work week of inhaled air averaged over a 7 day period (Stokinger and Woodward, 1958). A separate standard has been established for 1,1-DCE, but the TLV does not distinguish between the two isomers of 1,2-DCE.

The standard for 1,2-DCE was established on the basis of no measurable effects on growth, mortality, organ and body weights, hematology, clinical chemistry, and gross and microscopic pathology at doses of up to 4,000 mg/m³ for 6 months in rats, rabbits, guinea pigs and dogs (ACGIH, 1977). However, more recent data (Freundt, et al. 1977) indicate that 16 weeks' exposure at the TLV of 790 mg/m³ of trans-1,2-DCE produces histological evidence of fatty degeneration of the liver in rats.

In the case of 1,1-DCE, the standard was established primarily on the basis of the work of Prendergast, et al. (1977) who observed increased mortality as a result of continuous exposures of guinea pigs and monkeys to 61 mg/m³ and above for 90 days. Liver and kidney pathology were con-

sistently only observed at 189 mg/m³. As can be seen these industrial TLV's allow very little safety factor for sensitive populations. Recent data suggesting that 1,1-DCE is carcinogenic in mice (Maltoni, 1977) have not yet been taken into account.

Basis and Derivation of Criterion

The use of inhalation data assumes an 8-hour day, time-weighted average, occupational exposure in the work place with workers inhaling the toxic substance throughout such period. Exposures for the general population should be considerably less. Such worker-exposure inhalation standards are inappropriate for the general population since they presume an exposure limited to an 8-hour day, an age bracket of the population that excludes the very young and the very old, and a healthy worker prior to exposure. Ingestion data are far superior to inhalation data when the risks associated with the food and water of the water environment are being considered. Recent data suggest that the ACGIH estimate of noncarcinogenic risks resulting from exposure to the DCE's may approximate effect levels for both 1,1-DCE and isomers of 1,2-DCE. Additionally, it is recognized that the ACGIH standards apply primarily to healthy adult worker populations and do not incorporate safety factors for sensitive populations. In order to provide a wider margin of safety, calculations of acceptable concentrations of DCE's in drinking water as proposed by Stokinger and Woodward (1958) include a safety factor of 100 and are illustrated below:

$$1,1\text{-DCE} \quad \frac{40 \text{ mg/m}^3 \times 50 \text{ m}^3/\text{week} \times 0.40^*}{7 \text{ days/week} \times 100^{**}} = 1.14 \text{ mg/day}$$

$$\text{cis- and trans-1,2-DCE} \quad \frac{790 \text{ mg/m}^3 \times 50 \text{ m}^3/\text{week} \times 0.40^*}{7 \text{ days/week} \times 100^{**}} = 22.6 \text{ mg/day}$$

*Estimated coefficient of absorption via inhalation vs. ingestion.

**Safety factor for sensitive populations.

The safety factor of 100 was based upon the rationale of the National Academy of Sciences for non-carcinogenic substances where limited human data and valid animal studies exist (NAS, 1977). The absorption coefficient is based on the data of Monster, et al. (1976) for the related compound, trichloroethylene. Assuming a 2 liter daily consumption of drinking water, concentrations of 1,1-DCE should be limited to 0.6 mg/l (1.14 mg/day/2 liters) and the cis- and trans-1,2-DCE to 11 mg/l (22.6 mg/day/2 liters) on the basis of non-carcinogenic risks.

Under the Consent Decree in NRDC v. Train, criteria are to state "recommended maximum permissible concentrations (including where appropriate, zero) consistent with the protection of aquatic organisms, human health, and recreational activities." 1,1-Dichloroethylene is suspected of being a human carcinogen. Because there is no recognized safe concentration for a human carcinogen, the recommended concentration of 1,1-dichloroethylene in water for maximum protection of human health is zero.

Because attaining a zero concentration level may be infeasible in cases and in order to assist the Agency and States in the possible future development of water quality regulations, the concentrations of 1,1-dichloroethylene corresponding to several incremental lifetime cancer risk levels have been estimated. A cancer risk level provides an estimate of the additional incidence of cancer that may be expected in an exposed population. A risk of 10^{-5} for example, indicates a probability of one additional case of cancer for every 100,000 people exposed, a risk of 10^{-6} indicates one additional case of cancer for every million people exposed, and so forth.

In the Federal Register notice of availability of draft ambient water quality criteria, EPA stated that it is considering setting criteria for 1,1-dichloroethylene at an interim target risk level of 10^{-5} , 10^{-6} or 10^{-7} as shown in the table below.

<u>Exposure Assumptions</u> (per day)	<u>Risk Levels and Corresponding Criteria (1)</u>			
	<u>0</u>	<u>10^{-7}</u>	<u>10^{-6}</u>	<u>10^{-5}</u>
2 liters of drinking water and consumption of 18.7 grams fish and shellfish. (2)	0	0.013 $\mu\text{g/l}$	0.13 $\mu\text{g/l}$	1.3 $\mu\text{g/l}$
Consumption of fish and shellfish only.	0	0.21 $\mu\text{g/l}$	2.1 $\mu\text{g/l}$	21 $\mu\text{g/l}$

- (1) Calculated by applying a modified "one-hit" extrapolation model described in the Methodology Document to the animal bioassay data presented in Appendix I and in Table 12. Since the extrapolation model is linear at low doses, the additional lifetime risk is directly proportional to the water concentration. Therefore, water concentrations corresponding to other risk levels can be derived by multiplying or dividing one of the risk levels and corresponding water concentrations shown in the table by factors such as 10, 100, 1000, and so forth.
- (2) Approximately 6 percent of the 1,1-dichloroethylene exposure results from the consumption of aquatic organisms which exhibit an average bioconcentration potential of 6.9-fold. The remaining 94 percent of 1,1-dichloroethylene exposure results from drinking water.

Concentration levels were derived assuming a lifetime exposure to various amounts of 1,1-dichloroethylene, (1) occurring from the consumption of both drinking water and aquatic life grown in waters containing the corresponding 1,1-dichloroethylene concentrations and, (2) occurring solely from consumption of aquatic life grown in the waters containing the corresponding 1,1-dichloroethylene concentrations. Because data indicating other sources of 1,1-dichloroethylene exposure and their contributions to total body burden are inadequate for quantitative use, the figures reflect the incremental risks associated with the indicated routes only.

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Appendix I

Derivation of Criterion for 1,1-Dichloroethylene (DCE)

Maltoni exposed mice to 25 ppm via inhalation for 4 hrs/day, 4.5 days/week for 1 year. The interim report summarized the results after 82 weeks or 1.64 years. The control males had a kidney adenocarcinoma incidence of 0/126 and the treated males had an incidence of 16/98. The fish bioaccumulation factor is 6.9.

The parameters of the extrapolation model are:

$$\begin{aligned} nt &= 16 & le &= 52 \text{ weeks} \\ NT &= 98 & d &= 25 \times 4/24 \times 4.5/7 = 2.6786 \text{ ppm} \times 0.4 \\ & & &= 1.0714 \end{aligned}$$

$$nc = 0 \quad w = 0.025 \text{ kg (not needed)}$$

$$NC = 126 \quad L = 90 \text{ weeks}$$

$$Le = 82 \text{ weeks} \quad R = 6.9$$

$$D = 1.0714 \times 52/82 = 0.67944 \text{ ppm}$$

$$t = 82/90 = 0.91111; t^3 = 0.75633$$

$$Dt^3 = 0.51388 \text{ ppm}$$

$$\begin{aligned} B &= 1/.51388 \ln[(1-0/126)/(1-16/98)] \\ &= 0.34686 \text{ (in terms of risk per ppm)} \end{aligned}$$

For air exposures in units of ppm, no species conversion factor is necessary. Therefore for humans the concentration, X, giving a risk, R, of 10^{-5} is $X = R/B = 10^{-5}/0.34686 = 2.8830 \times 10^{-5} \text{ ppm}$.

5800

510

Since the air concentration corresponding to 1 ppm of DCE is $4 \times 10^3 \mu\text{g}/\text{m}^3$ and since people breathe an average of $24 \text{ m}^3/\text{day}$ of air, the daily intake of DCE that would result in a lifetime risk of 10^{-5} is:

$$\begin{aligned} 2.8830 \times 10^{-5} \text{ ppm} \times 4 \times 10^3 \mu\text{g}/\text{m}^3 \text{ per ppm} \times 24 \text{ m}^3/\text{day} \\ = 2.7677 \mu\text{g}/\text{day} \end{aligned}$$

If it is assumed that the fraction of the DCE intake reaching the target site is the same whether it is administered via inhalation or ingestion of water and fish, a daily DCE intake of $2.768 \mu\text{g}$ through water and fish alone would also cause a cancer risk of 10^{-5} . The water concentration giving this intake is:

$$\begin{aligned} C &= \frac{2.7677 \mu\text{g}/\text{day}}{(2 + 6.9 \times .0187) \text{ liters}/\text{day}} \\ &= 1.2999 \mu\text{g}/\text{l} \\ &1.3 \mu\text{g}/\text{l} \end{aligned}$$